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FUELS AND FIRE ECOLOGY: A GUIDE FOR LAND MANAGERS IN THE SOUTHWESTERN UNITED STATES

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FINAL REPORT

Fuels and Fire Ecology: A Guide for Land Managers in the Southwestern United States

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Guy R. McPherson

Introduction

The purpose of this monograph is to integrate and synthesize scientific information on the fire ecology of major forest and grassland types within the context of fire management in the southwestern United States, especially in Arizona and New Mexico. It responds to the need for a consolidated and concise synthesis of fire ecology, fuels management, and fire management at regional and local scales in the southwestern United States. This monograph matches and responds to a critical information gap: The need for credible information about fuel-management treatments was identified as the highest priority by researchers and managers who participated in a recent series of workshops in the western United States (White 2004). As part of this large gap, land managers, policy makers, fire management officers and fuels specialists need information about historical and contemporary fire regimes. In addition, this document is intended to facilitate appropriate application of ecological information in the development of monitoring plans and the management of contemporary ecosystems. The monograph is organized by coarse-scale ecosystems, and it provides a general synthesis intended to aid site-specific fire management and project-level planning. The intended audiences are researchers and land managers who will apply this information to project-level planning and site-specific management of fuels and fire.

The information contained in this document should facilitate the preparation of land management and fire management plans and management strategies for wildland-urban interfaces. This information will serve as an essential reference in the development of fire management units and in National Environmental Policy Act (NEPA) decision documents. Prior to preparation of this document, critical information was spread throughout myriad resources. Because this monograph is based on broadly defined ecosystems rather than political boundaries, it should prove useful to several land management agencies and organizations and assist in cross-boundary collaboration.

This monograph is organized around the major ecosystems in the southwestern United States, and it incorporates research about fire ecology that is directly applicable to fuels and wildland fire management. It is based on a comprehensive review of relevant literature. When possible, it relies on studies from the southwest, but also incorporates information from similar ecosystems in other regions. It considers the following ecosystems within Arizona and New Mexico: spruce-fir forest, mixed-conifer forest, ponderosa pine forest, evergreen oak woodland, pinyon-juniper woodland, and southwestern grassland. Savannas dominated by oak, juniper, and mesquite are described within the context of adjacent woodlands and grasslands. Although this document is brief, it attempts to provide insights for management; however, progressive managers will use site-specific information to develop management plans (*sensu* McPherson 1997, McPherson and Weltzin 2000). Consideration of riparian systems is beyond the scope of this monograph, and response of these systems to fire has been described recently (2003, *Forest Ecology and Management* special issue, volume 178).

Format

For each ecosystem, this document describes the setting and species composition, historical fire regimes and their consequences, and fire effects on common species; adjacent ecosystems typically share common species, which engenders necessarily redundant information among broad-scale ecosystems in this latter section. The document briefly summarizes major changes in ecosystem attributes since Anglo settlement and provides suggestions for land managers to use in planning fire management projects given these altered conditions. Consideration of the altered state of a given ecosystem is critical for planning fire and fuel management projects. The rarity of some species, introduction of nonnative species, ever expanding human population, and geographic fragmentation are among the many factors that greatly confound efforts to manage fuels and restore community structure and function in remaining wildlands of the southwest.

Fire effects on individual species

Native species in all the major ecosystems described here have evolved adaptations to periodic fires. In the American Southwest, the historical prevalence of fire ensures that even those species that seem most intolerant of fire have evolved in the presence of recurrent fires, with the exception of species in the Mojave and Sonoran Deserts, which rarely burned before Anglo settlement (Wright and Bailey 1982; Steuter and McPherson 1995). Adaptations to fire are many and diverse, and include escape (e.g., distributions limited to rocky areas where fire rarely occurred), tolerance (e.g., thick bark), and rapid recruitment in post-fire environments (e.g., widely dispersed seeds and ability to establish in open environments). In addition, the online Fire Effects Information System (http://www.fs.fed.us/database/feis/) provides considerable information about the short-term response of individual species to fire. As a result, this document summarizes the response to fire of dominant species within each coarse-scale ecosystem, but it does not provide detailed information about most other species in any of the types. Non-dominant species on any particular site are rarely inventoried, much less monitored on a recurring basis; as a result, we know very little about most species in the region. Thus, the response of most species to various fire regimes remains unknown.

Historical fire regimes and their consequences

The interactions of fire with climatic factors such as drought influence the composition, structure and arrangement of natural communities. In addition to fire frequency, the severity, patch size and shape of areas burned, and the seasonality of the burn, can greatly influence community attributes (Pickett and White 1985, Agee 1998). That is, native species have developed adaptations to fires that occur at a particular frequency, season, and extent. Further, variability of these attributes may influence fire's effect on plant and animal communities (Aplet and Keeton 1999). Consider two scenarios, both with an average fire interval of 20 years. In the first scenario, fires occur on a regular interval of 20 years. In the second scenario, the interval between fires was 4, 21, 60, and 4 years. The ecological effect of these two fire regimes might be quite different, especially for species such as aspen that would be maintained at the regular 20-year intervals, but be taken over by conifers during the 60-year fire-free interval in the second scenario. Appropriate contemporary management keeps variability foremost in mind. For example, variability in time between fires was substantial relative to mean fire frequency: A stand of ponderosa pine trees that burned every 11.8 years before Anglo settlement undoubtedly experienced fire-free intervals as short as 1 or 2 years and as long as 30 years.

Historical fire regimes in forests typically have been evaluated with dendrochronological analyses. This approach is useful as a reference tool, particularly when used to compare ecosystems. For example, fires certainly were more common in ponderosa pine forests of the American Southwest than in nearby mixed-conifer forests, which is consistent with dendrochronological interpretations. Fire-scar records based on dendrochronological analyses are useful in a relative sense, if not an absolute one. Further, the literature is replete with results of such analyses, and is generally lacking in other kinds of evidence. Therefore, this document summarizes the results of several dendrochrological analyses.

There is disagreement about how well dendrochronological analyses represent historical frequency of fires. Consider, for example, western forests dominated by ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.). With respect to fire history, these are the best-studied forests in the world. According to myriad dendrochronological analyses, ponderosa pine forests have composite fire intervals of 2-25 years. Techniques that account for the biases associated with dendrochronological methods indicate that these forests probably burned every 22-308 years (Baker and Ehle 2001, 2003). However, researchers have begun to address concerns raised by Baker and Ehle, and emerging data suggest that sampling “recorder” trees (trees with multiple fire scars) instead of sampling at random or on a grid biases the fire frequency only slightly: Standard techniques failed to record only fires smaller than 20 acres, albeit on sites that were admittedly unusual because fire regimes had been minimally interrupted and timber had never been harvested (Fulé and others 2003a; Heyerdahl and others [2001] presented similar results for the Blue Mountains of eastern Oregon and southeastern Washington). It is generally assumed that large fires are important drivers of ecosystem change, at least relative to small fires; nonetheless, a tight focus on large fires in the historical record masks considerable spatial heterogeneity produced by small fires, and relying on mean fire interval of large fires as a basis for reintroducing fire into these systems will produce systems that are more homogeneous than historical systems (e.g., Agee 1998). The use of dendrochronology to determine fire frequency clearly requires further study, and the results of these analyses should be interpreted with considerable caution. However, they generally illustrate a summer-dominated fire regime, high variability in fire frequency over time, and fire regimes with a combination of surface fires and stand-replacing fires (e.g., citations in the following paragraph).

Again focusing on western ponderosa pine communities because they are so well studied, some researchers have concluded that historical fire regimes in ponderosa pine forests did not include periodic stand-replacing fires. A contrary conclusion is reached, albeit tentatively, by Cooper (1960) and subsequently supported by considerable evidence (e.g., White 1985; Swetnam and Baisan 1996b; Shinneman and Baker 1997; Brown and others 1999; Baker and Ehle 2001; Huckaby and others 2001; Baker and Ehle 2003; Ehle and Baker 2003). In fact, the historical fire regime was not comprised exclusively of surface fires, as is widely reported or suggested (e.g., the National Fire Plan; Swetnam and Betancourt 1990; Swetnam and Baisan 1996b; Touchan and others 1996; Covington and others 1997; Fulé and others 1997; Swetnam and Betancourt 1998; Swetnam and others 1999; Covington 2003; Friederici 2003). Suppression of crown fires, though not entirely possible, would actually move the fire regime outside its historical range of variability (Ehle and Baker 2003; Pierce and others 2004). Reintroducing stand-replacing fires in these systems in the name of ecological restoration likely will not reduce hazard to human lives or property, and creating new fire regimes comprised of frequent surface fires does not represent ecological restoration. The very frequent fires often deemed appropriate for ecological restoration of these systems simply cannot be justified on the basis of restoration: such attempts are not only inconsistent with ecological restoration, they are likely to be futile (Veblen 2003; Ehle and Baker 2003), especially in the face of regional temperature regimes that are becoming eerily similar to those experienced during the Medieval Climatic Anomaly of ca. 650-1,050 years ago (Pierce and others 2004).

While not serving as a complete and unbiased record of past fires, fire-scar evidence can be regarded as an index of fire occurrence (Johnson and Gutsell 1994; Minnich and others 2000; Baker and Ehle 2001; Veblen 2003). Yet a complete and unbiased record of past fires underpins accurate restoration of the fire regime, hence true ecological restoration (Veblen 2003).

Notwithstanding these important points about restoring ecological processes in western ponderosa pine communities, two additional points bear mention. First, the understanding we lack about western ponderosa pine communities is minor compared to the paucity of knowledge about other ecosystems. Disagreements about fire history and the appropriate use of fire will continue to plague management of wildland ecosystems. However, the absence of knowledge is insufficient justification to take action: Even the absence of action represents management, and neglect rarely is appropriate in western systems that have been long neglected. Second, management of these systems need not, and should not, be guided strictly by restoration. Rather, managers must consider social, economic, and political issues when making decisions about wildland ecosystems.

From a strictly theoretical standpoint, restoration is almost never possible. Restoration of community structure and ecosystem processes to conditions prevalent at any time in the past is a lofty goal that, given the complexity of natural systems, simply cannot be achieved on most sites. Terms such as "rehabilitation," "reclamation," "repair," and "revegetation" are more appropriate descriptors than "restoration" for management actions that attempt to "heal" or "conceal" anthropogenic disturbances (National

Research Council 1974, Society for Ecological Restoration 1994, Aronson and others 1993, Whisenant 1999).

Although complete ecological restoration may not be achievable for most ecosystems, it nonetheless offers a goal toward which progress should be measured. We should not abandon efforts to restore ecosystems merely because the most Herculean efforts will fall short. The appropriate metric likely is some distance, as yet undefined, from the goal of complete restoration; progress toward this goal probably is best evaluated in relative, rather than absolute, terms. As such, management efforts geared toward retention of high levels of native-species biological diversity should place high priority on areas that are most similar to the desired condition at the expense of areas that are dissimilar from desired conditions (e.g., historical "reference" condition). A focus on systems that closely resemble historical "reference" conditions will require far fewer cultural inputs, and therefore will be less expensive, than a focus on systems that are vastly dissimilar from "reference" conditions. For maximum impact, maintenance of sites that resemble historical "reference" conditions should focus on ecological processes, not individual species.

Fire management in altered contemporary ecosystems

Although individual fire managers have no direct control over natural ignition sources and weather, they must consider the relative roles of weather, fuels, and ignition on a site-specific basis. For example, ongoing human activities are causing imperilment of species, continued fragmentation of ecosystems and introduction of nonnative species, which may substantially impact the ability of managers to affect the fire hazard or reintroduce fire on a particular site. Indeed, the consequences of species losses, fragmentation, and biological invasions are likely to constrain the reintroduction of fire or the mitigation of fire hazard to a greater extent than manipulation of fuels, much less determination of the historical fire regime.

Fire hazard reduction and maintenance of biological diversity are achievable goals for some ecosystems and sites. The goal of reducing fire hazard can be quantified and evaluated objectively. Similarly, maintenance of biological diversity is an important and measurable goal. Thus, the remainder of this document assumes that the specific objectives of management are tied to the general goals of (1) reducing the incidence of fires that are difficult to control, particularly in areas populated by humans (i.e., hazard reduction) and (2) maintaining high levels of native biological diversity. The first of these goals is particularly relevant to property and human lives, and it also influences the second goal. Maintenance of biological diversity is important because present and future generations of humans depend on a rich diversity of life for survival and civilization. As architects of the extinction crisis currently facing planet Earth, we have a responsibility to future generations of *Homo sapiens* and to non-human species to retain as much biological diversity as possible. The substantial economic cost of maintaining high levels of biological diversity likely will pale in comparison to the costs of failing to do so.

Recognition that native species in the American Southwest (except in those found in deserts) evolved in concert with periodic fires leads to two general conclusions: (1) These native species have developed adaptations to fires that occur at a particular frequency, season, and extent; and (2) maintenance or reintroduction of the fire regimes with which these species evolved should assume high priority for those interested in maintaining high levels of biological diversity. A corollary to the first conclusion is that classification of native species along a gradient of adaptation to fire is simplistic and potentially misleading. All native species are "adapted" to recurrent fires, and classifying some as more tolerant than others suggests that fire is "good" for some species and "bad" for others. A more appropriate view is that recurrent fires, at the appropriate frequency, season, and extent, are part and parcel of these ecosystems. A corollary of the second conclusion is that reintroducing ecological processes rather than targeting specific stand structure should be a relatively efficient and comprehensive strategy for retaining native species in extant ecosystems, albeit with two caveats about contemporary fire management.

First, contemporary fires often are unlike the fires with which species evolved and communities developed. A century of fire suppression, along with the introduction of nonnative species and regional warming, has changed the mix of species and wildland fuels in many of these systems. In addition to new actors on the evolutionary stage, warmer weather and relatively homogeneous fuels characterize fires that occur in areas formerly typified by cooler weather and heterogeneous fuels. Fires in these "new" systems may favor nonnative species to a greater extent than they favor native species, thus reducing biological diversity. If a nonnative species is prevalent or potentially prevalent on a particular site, managers should exercise caution when attempting to reintroduce fire.

In addition to these concerns about anthropogenic alterations in fuels and species composition, a second caveat is warranted. Anthropogenic activities have fragmented landscapes, and this fragmentation has blocked routes of dispersal. Former metapopulations no longer exist as sources for small, isolated populations, which may amplify the effects of small fires. For example, fire-induced floods lead to substantial movement of sediment, and this sediment chokes small streams and extirpates local populations of aquatic organisms. Such events have undoubtedly occurred with some regularity during the last ten thousand years. But only during the last century has human-caused fragmentation eliminated all potential paths of reoccupation for many such sites. Previously interconnected populations of aquatic organisms are now isolated and are therefore susceptible to the sediment freed by fires. Because of change in species composition, fuel loads, and metapopulation dynamics, contemporary fires may extinguish small, isolated populations of many species in southwestern ecosystems.

Management and Hazard Reduction

Ascertaining the historical fire regime is a difficult task, in part because it requires site-specific data that usually are absent or lacking in depth. At the very least, considerable information about fire history is required. The common approach to acquiring these data employs dendrochronological analyses; these are occasionally augmented with notes from General Land Office Surveys or historical photographs.

An alternative to this comparative structure-based approach is to reintroduce processes that were prevalent before Anglo settlement and that have been subsequently interrupted. The dominant ecological process in western forests before Anglo settlement was fire (e.g., Ehle and Baker 2003), and its occurrence and spread have been vastly reduced. With respect to maintenance of high levels of native-species biological diversity, it is particularly appropriate to reintroduce fires into southwestern ecosystems. Important questions focus on the season and frequency with which fires should be reintroduced, and these issues are discussed within the context of each broad-scale ecosystem. We know enough about both elements to reintroduce fire into these systems. Societal, political, managerial, and logistical concerns likely will continue to constrain reintroduction of fires to a far greater extent than ecological knowledge.

Fire Ecology and Fuels Management of Major Fuel Types

Spruce-fir forest

Spruce-fir forests cover about 20,000 acres in Arizona and New Mexico, primarily at elevations above 8,000 feet. These forests correspond to the southwestern spruce-fir type of Kuchler's (1964) classification system. With respect to elevation, they are the highest ecosystems in the southwestern United States, and they typically occur slightly above mixed-conifer forests with conifers that are dominant at slightly warmer and drier climates. These forests probably average between 25 inches and 40 inches of annual precipitation, although climatic records are conspicuously absent atop the mountains where these forests are found (Brown 1982, pp. 37-38).

Spruce-fir forests are uncommon in the American Southwest, yet they have received considerable attention from the scientific community (e.g., Alexander 1974; Jones 1974; Dye and Moir 1977; Moir and Ludwig 1979; Alexander 1987; Stromberg and Patten 1991; Dyer and Moffett 1999; Fulé and others 2003a). This attentiveness to a relatively uncommon ecosystem probably results in part from the high biological diversity these systems contribute to the region. Spruce-fir forests are the highest-elevation ecosystems in the region and, as the pinnacle ecosystems in southwestern "sky islands," they harbor several rare taxa [including the federally Endangered Mt. Graham red squirrel (*Tamiasciurus hudsonicus grahamensis*) and the federally Threatened Mexican spotted owl (*Strix occidentalis lucida*)]. Because of the moisture and temperature relations associated with their topographic position, these forests also serve as a bellwether for regional climate change.

Common tree species include Engelmann spruce (*Picea engelmannii* Parry), subalpine fir [*Abies lasiocarpa* var. *lasiocarpa* (Hook.) Nutt.] and its southern variety, corkbark fir [*Abies lasiocarpa* var. *arizonica* (Merriam) Lemm.], and quaking aspen (*Populus tremuloides* Michx.). Scattered individuals or small stands of blue spruce (*Picea glauca* Engelm.) are occasionally found, especially in riparian areas. White fir [*Abies concolor* (Gord. & Glend.) Lindl.] and Douglas-fir [*Pseudotsuga menziesii* var. *glauca*

(Breissn.) Franco] commonly occur after stand-replacing disturbances and may persist for several hundred years, especially on relatively dry sites. New Mexico locust (*Robinia neomexicana* Gray) occurs as a small understory tree.

Fire effects on common trees

Individual Engelmann spruce and blue spruce trees are poorly adapted to fire. Few individuals establish in rocky refuges protected from fire, and the thin bark, unprotected buds, shallow roots, persistent dead lower limbs, and low-hanging, layered branches that trap heat predispose these trees to high mortality even from low-intensity surface fires. In cool, moist areas, both species establish readily after stand-replacing disturbances. Spruce trees are slow to establish on drier sites such as southern exposures or near the low-elevation limit of this ecosystem. Since blue spruce typically occupies moist areas, delayed establishment on xeric sites is particularly obvious for Engelmann spruce, which may require an overstory canopy under which to establish (e.g., from the more xeric species, white fir or Douglas-fir; Patten 1963).

Like Engelmann spruce and blue spruce, both varieties of subalpine fir are very susceptible to fire. These taxa of spruce and fir are widely acknowledged to be among the least likely to survive fires in North America (Wright and Bailey 1982). Also similar to spruce trees, the northern variety of subalpine fir often colonizes after stand-replacing disturbance, though the conifer seedlings may be obscured by stems of quaking aspen. In contrast to the northern variety of subalpine fir, corkbark fir usually is not among the initial colonizers, a difference that may result from the larger seeds and later age of reproductive maturity for the latter variety (Alexander and others 1984; Stromberg and Patten 1991). Corkbark fir typically recruits 80-150 years after initial colonization by other trees on cool, moist sites and up to 250 years after colonization by other trees on more xeric sites (Stromberg and Patten 1991).

Pure stands of quaking aspen rarely burn except during prolonged droughts because aspen leaves and standing trees have high moisture content (Wright and Bailey 1982; Brown and Simmerman 1986; Rothwell and others 1991). Aspen stands are relatively rare in the southwestern United States. Rather, quaking aspen commonly occurs as scattered individuals or rootstocks within stands dominated by conifers. In these settings, the aboveground portions of quaking aspen trees usually are killed by fires, even ones of low to moderate intensity (DeByle 1985; Jones and DeByle 1985). Quaking aspen is the archetypical fire-adapted species in these systems: It resprouts vigorously and grows quickly after fire, such that it appears on first inspection to be the only tree present (Schier and others 1985; Perala 1990). The species has been reduced considerably in western North America since Anglo settlement, a change that has been attributed to reduction of stand-replacing disturbances, along with browsing by increasingly large herds of large ungulates (Shepperd 1986; DeByle and others 1987). Quaking aspen stems usually are short-lived in the American Southwest, and the species is very shade-intolerant (Perala 1990). As a result, seemingly monotypic stands of quaking aspen give way to dominance by conifers within 30-60 years.

White fir is not resistant to fire until it reaches a diameter of about 30 cm because it has thin, resinous bark and layered, low-growing branches that trap heat and serve as ladder fuels into the crown (Wright and Bailey 1982; Laacke 1990). As a result, most small trees are easily killed by even low-intensity surface fires. As trees age, their bark thickens and some of the lower branches die and fall away, thus enhancing the resistance of white fir to surface fires. However, even large trees retain some lower branches, and they have relatively shallow roots and often are covered with lichens, such that resistance to surface fires is moderate (Bradley and others 1991). Resistance is very low to the stand-replacing crown fires that typify southwestern spruce-fir forests. After stand-replacing fires, white fir establishes via small, winged seeds that are easily dispersed by wind. White fir trees then grow relatively slowly, but they may persist for one to three centuries before giving way to more shade-tolerant species. Two large, stand-replacing fires within a period shorter than the reproductive age of white fir (ca. 40 years, according to Laacke 1990) may cause populations of the species to be eliminated from large areas for centuries or millennia.

Douglas-fir is the most fire-resistant tree in this ecosystem. As a mature tree, it has thick bark that provides ample insulation from surface fires. However, even large trees are susceptible to the stand-replacing fires that characterize this ecosystem. Like white fir, this species establishes after stand-replacing fires via seeds that are wind-dispersed (Owston and Stein 1974; Hermann and Lavender 1990). Most recruitment occurs on relatively xeric sites; more mesic sites are occupied by one or more of the species listed above. Individual trees then grow rapidly and they may persist for several centuries. Similarly to white fir, successive stand-replacing fires over a large area may cause populations of this species to be extirpated from large areas for very long periods of time.

New Mexico locust forms dense thickets beneath some conifer stands throughout the region. Fires, even surface fires of low intensity, easily kill stems. However, the species resprouts readily when top-killed. In addition, seeds require scarification for germination. Thus, fires enhance sprouting of top-killed trees and recruitment of seedlings. Conifers typically overtop New Mexico locust when they reach 15 to 20 years of age.

Fire regimes and their consequences

The typical fire regime before Anglo settlement included stand-replacing fires of relatively high intensity every few hundred years (Grissino-Mayer and others 1995; Fulé and others 2003a). Before Anglo settlement, these fires typically were ignited by lightning during late June and early July thunderstorms at the leading edge of the southwestern "monsoon." In the absence of fire suppression, these fires burn until they are doused by heavy "monsoonal" storms; during years of little summer precipitation, they may burn until snow flies.

Periodic fires may be required to retain spruce as a significant component of these forests. Shade-intolerant spruce trees often are replaced by subalpine fir or corkbark fir in the long-term absence of fire (Peet 1988). However, low adult mortality and long life-span of Engelmann spruce allow the species to persist for 500 years or more in the absence of fire (Shea 1985). Nonetheless, fire was a recurrent phenomenon and presumably an integral component of forest dynamics in these systems before Anglo settlement (Stromberg and Patten 1991).

Given the long fire-return intervals that characterized these forests before Anglo settlement, it seems reasonable to assume that a century of fire suppression has had minimal impact on disturbance dynamics or stand structure. Thus, the contemporary fire regime probably is very similar to the pre-settlement regime. Prescribed fires have not been attempted in these forests. Considering the susceptibility of dominant tree species to fires and the fuel and weather conditions associated with historical and contemporary fires, introducing prescribed fires into spruce-fir forests seems unlikely and unwarranted.

Although the fire regime apparently has been largely unaffected by Anglo settlement, the impacts of contemporary fires probably will differ from those of pre-settlement fires. Specifically, regional warming and drying as a result of anthropogenic global warming probably affect tree recruitment patterns. Given the tendency for Engelmann spruce to establish slowly on relatively xeric sites, and the 80- to 250-year lag in establishment of corkbark fir, it is increasingly likely that stand-replacing fires will cause spruce-fir forests to be replaced by forests dominated by white fir or Douglas-fir. Although a warmer, drier climate has been gradually replacing a cooler, wetter climate since the advent of the Holocene some 11,000 years ago, the dramatic acceleration of this pattern during the last century portends rapid changes in stand structure of these high-elevation forests. Rapid warming and drying likely will elevate the likelihood of fire occurrence and intensity in these forests; this pattern already is evident in some North American coniferous forests (Wotton and others 2003), and models indicate the pattern will soon spread to other such forests (Baker 2003; Flannigan and others 2003; Miller 2003). Warming and drying further threatens these forests by elevating moisture stress, thereby weakening individual trees and predisposing them to attack by insects such as the native western spruce budworm (*Choristoneura occidentalis*), the native spruce beetle (*Dendroctonus rufipennis*), and several native and nonnative species of aphid (genera *Adelges*, *Cinara*, and *Pineus*). The resulting increase in dead wood threatens to exacerbate the increased probability of fire occurrence and spread produced more directly by anthropogenic warming and drying.

Land tenure is predominantly U.S. Forest Service, and structures are uncommon in this ecosystem. The primary exception is commercial development associated with ski areas and occasional telescopes and radio towers. Dense stands of spruce and fir trees intermingled with cabins and commercial enterprises—usually constructed from wood—represent nightmare scenarios of fire prevention and control. These ecosystems burned periodically before Anglo settlement, and the fire regime was (and is) controlled by weather. The inevitable high-intensity fires that characterize these forests simply shrug off human attempts at suppression.

Mechanical treatment of fuels

Removal of ladder fuels may engender the temporary replacement of stand-replacing crown fires with surface fires of lower intensity, and sufficiently aggressive pruning and thinning may substantially reduce fire hazard in these forests. The persistence of such fuel-reduction treatments has not been evaluated; considering the rapid development of the fine fuels responsible for fire spread, mechanical fuel-reduction treatments probably do not persist beyond five years, a conclusion that is consistent with a worldwide review of evidence on hazard reduction (Fernandes and Botelho 2003).

Mechanical removal of fuels is economically expensive, especially in remote areas. Nonetheless, expensive and transient fuel-reduction treatments may be justified by the significant threat of fire in the absence of treatments and the associated values at risk (especially populations of people or rare species). Ultimately, however, mitigating for the hot and dry climatic conditions that predispose these forests to high-intensity crown fires is a battle not likely to be won by manipulating fuels: Weather drives fire occurrence in these forests, and the weather increasingly favors fires. That is, fire spread in these forests is constrained largely by fuel moisture (Rollins and others 2002), which indicates that attempts to manipulate fuels likely will not reduce the incidence of fires.

Those who propose mechanical reduction of fuels in spruce-fir forests should bear in mind the final sentence of Richard Feynman's (1986) report on the tragic explosion of the Challenger space shuttle: "For a successful technology, reality must take precedence over public relations, for Nature cannot be fooled."

Treating fuels with livestock

Herbaceous plants contribute little to the fuel load. Rather, nearly all the fuel is downed woody material, low-hanging dead branches, and conifer leaves dried by hot, dry weather. Thus, removal of fuel via livestock grazing has little potential for reducing fire hazard.

Recommended treatments

No treatments are recommended to manage fuels in this type except wildland fire use (i.e., wildland fires burning within designated conditions). High-intensity, stand-replacing fires that likely would enhance biological diversity are too dangerous to implement as prescribed fires, given contemporary technology. Livestock are ineffective managers of fuel, and mechanical treatments represent a transient, expensive, and likely ineffective subsidy on economic activity in an inappropriate location.

Mixed-conifer forest

Mixed-conifer forests are found at sites drier than spruce-fir forests and moister than those dominated by ponderosa pine (*Pinus ponderosa* var. *scopulorum* P. & C. Larson). This document combines the spruce-fir-Douglas-fir and pine-Douglas-fir types of Kuchler's (1964) classification into the general category of mixed-conifer forest.

These forests are found on about two million acres in Arizona and New Mexico (Jones 1974), and typically are found at elevations between 6,500 feet and 9,000 feet. Although climatic records are uncommon, average annual precipitation apparently varies between 20 inches and 30 inches (Brown 1982, pp. 42-48).

Common tree species include Douglas-fir, southwestern white pine (*Pinus strobiformis* Engelm.), ponderosa pine, white fir, and quaking aspen, often in that order of abundance. New Mexico locust is a common understory tree. Pinyon-juniper woodlands, which also may be classified as mixed-conifer forests, typically are found on drier sites than stands of ponderosa pine and are described in later sections.

Fire effects on common trees

As a mature tree, Douglas-fir has thick bark that provides ample insulation from surface fires. However, even large trees are susceptible to the stand-replacing fires that periodically occur in this ecosystem. This species establishes via seeds that are wind-dispersed (Owston and Stein 1974; Hermann and Lavender 1990). Most recruitment occurs on mineral soil exposed by disturbance, including fire that removes litter. Individual trees then grow rapidly enough that many individuals become relatively resistant to surface fires. Some trees of this species may persist for several centuries, until a stand-replacing fire kills virtually all trees. Successive stand-replacing fires over a large area may cause populations of this species to be extirpated for very long periods of time.

Southwestern white pine is moderately resistant to fire as a mature tree (Ahlstrand 1980; Dieterich 1983). Seedlings are very susceptible to fires, even fires of low intensity. As trees age, their normally thin bark thickens and some of the drooping lower branches die and fall away, thus enhancing the resistance of this species to surface fires. Recruitment into burned areas is scattered and inconsistent, at least in part because this species has large, essentially wingless seeds that are not well dispersed by wind (Perry 1991) and because it is relatively shade-tolerant. The irregular pattern of fires in these forests, coupled with the presence of southwestern white pine in cool, shaded areas, suggests that a few individuals escape most fires and then act as sources of seed for post-fire recruitment.

Ponderosa pine has many attributes that make it very fire-tolerant (Weaver 1967; Boldt and Van Deusen 1974; Schubert 1974; Dieterich 1979; Wright and Bailey 1982; Sackett 1984; Oliver and Ryker 1990). Adult trees have thick, platy bark; the spaces between plates dissipate heat, and the plates exuviate, thus protecting cambium. The relatively open canopy also allows heat to dissipate, thereby minimizing crown scorch. Also, buds are encased in a fascicle of needles that absorbs much heat before the well-insulated buds can be damaged. Ponderosa pine is deep-rooted, even at an early age. In addition to its considerable resistance to fire, ponderosa pine generally recruits readily into mixed-conifer forests after fire. Although ponderosa pine seeds are large and therefore not particularly well dispersed by wind, the relatively patchy character of fires in these forests ensures an adequate supply of seeds from adult trees near recently burned areas. Ponderosa pine seedlings establish readily in mineral soil in these relatively mesic forests. Height growth and diameter growth are rapid, such that trees become large and therefore fire-resistant relatively quickly. Seedlings are moderately susceptible to fire, with mortality strongly dependent on fire intensity. For example, most seedlings less than 4.5 feet tall survived "light" (and presumably low-intensity) surface fires on the Fort Apache Indian Reservation, about half survived "hot" (and presumably high-intensity) surface fires, and about one-fourth survived crown fires (Lindenmuth 1960). This combination of high recruitment, rapid growth, and high resistance makes ponderosa pine particularly well suited to surviving surface fires and establishing in the wake of crown fires.

White fir is not resistant to fire until it reaches a diameter of about 30 cm because it has thin, resinous bark and layered, low-growing branches that trap heat and serve as ladder fuels into the crown (Wright and Bailey 1982; Laacke 1990). As a result, even low-intensity surface fires easily kill most small trees. As trees age, their bark thickens and some of the lower branches die and fall away, thus enhancing the resistance of white fir to surface fires. However, even large trees retain some lower branches, and they have relatively shallow roots and often are covered with lichens, which makes these trees only moderately resistant to surface fires (Bradley and others 1991). Resistance to stand-replacing crown fires is very low. White fir seeds are easily dispersed by wind, but white fir trees typically establish in mixed-conifer forests only beneath other trees. While fir trees then grow relatively slowly but are protected from surface fires until they develop relatively thick bark. They may persist for several centuries until a stand-replacing fire kills them. Two large, stand-replacing fires within a period shorter than the reproductive age of white fir (ca. 40 years, according to Laacke 1990) may cause populations of the species to be eliminated from large areas for centuries or millennia.

Quaking aspen commonly occurs as scattered individuals or rootstocks within mixed-conifer forests because these sites tend to be too xeric to support relatively pure stands of this species. Even surface fires of moderate intensity typically kill the aboveground portion of quaking aspen trees (DeByle 1985; Jones and DeByle 1985). Quaking aspen resprouts vigorously and grows quickly after fire, but stems usually are short-lived and the species is very shade-intolerant (Perala 1990). As a result, quaking aspen gives way to dominance by conifers within 30-60 years.

Response of New Mexico locust to fire in mixed-conifer forest is similar to response in spruce-fir forest. That is, fires, even surface fires of low intensity, easily kill stems. However, the species resprouts readily and recruits from seeds. Thus, New Mexico locust trees sometimes form dense thickets that dominate local areas for 15-20 years before being overtopped by conifers.

Fire regimes and their consequences

Most mixed-conifer forests resulted from fire, and the relatively complex, mixed-severity fire regimes that create and maintain these forests produce a rich mosaic of trees across the landscape. Before Anglo settlement, fire intensity varied considerably within a stand, such that some areas did not burn, others burned as low-intensity surface fires, and other patches burned as high-intensity crown fires. Lightning during summer, especially during late June and early July, ignited virtually all fires; a few fires may have been lit by native Americans, although such ignitions are quite difficult to demonstrate conclusively (e.g., Seklecki and others 1996). This mixed-severity regime is undoubtedly responsible for the presence of trees with varying fire tolerances, and for intra- and inter-stand diversity in species composition. There is little question that fire was a recurrent phenomenon and presumably an integral component of forest dynamics in these systems before Anglo settlement. Dendrochronological analyses reportedly suggest mean surface-fire intervals of 3 to 20 years (Baisan and Swetnam 1990, 1995; Swetnam 1990; Grissino-Mayer and others 1995; Danzer and others 1996; Seklecki and others 1996; Swetnam and Baisan 1996a; Touchan and others 1996; Fulé and others 2002, 2003a, 2003b). Other analyses conclude the mean time between surface fires ranged from 30 to 200 years (Baker and Ehle 2001, 2003). Both sets of figures suggest that the surface-fire regime has been disrupted by anthropogenic activities, and indicate that surface fires should be reintroduced into these systems during the summer, when they occurred historically.

The contemporary fire regime has been affected by anthropogenic activities. The suppression of low-intensity surface fires during the last century has reduced fire spread appreciably. Fire spread has been further reduced by human-induced fragmentation. Occurrence of historically infrequent high-intensity crown fires has been reduced within landscapes and within stands. Decreased heterogeneity of fires within stands—specifically a reduction in patches that burn into the forest canopy—translates to more homogeneous stand conditions and further increases in fuel accumulation. The virtual absence of surface fires and the attendant reduction in patches of trees that "crown out" has allowed large woody fuels to accumulate. Along with expected rapid warming and drying, these factors likely have elevated the likelihood of stand-replacing fires in these forests.

Land tenure is predominantly U.S. Forest Service, and structures are uncommon in this ecosystem. The primary exceptions are a few small towns and commercial development associated with ski areas. Dense stands of conifer trees intermixed with cabins and commercial enterprises represent a nightmare scenario for fire prevention and control. These ecosystems burned relatively frequently before Anglo settlement, sometimes at high intensity. The occurrence of stand-replacing crown fires is strongly influenced by weather, and such fires are only minimally influenced by suppression efforts.

Mechanical treatment of fuels

Removal of ladder fuels may delay the onset of stand-replacing crown fires, thus maintaining a surface-fire regime characterized by low-intensity fires. The persistence of such fuel-reduction treatments has not been evaluated; considering the rapid development of the fine fuels responsible for fire spread, mechanical fuel-reduction treatments probably do not persist beyond five years, a conclusion that is consistent with a worldwide review of evidence on hazard reduction (Fernandes and Botelho 2003). Forest thinning contributed to increased, rather than decreased, fire damage in mixed-conifer forests in the Pacific Northwest, apparently because thinning increased the amount of fine woody material that served as fuel (Raymond and Peterson 2005).

High-intensity management of fuels may be economically expensive, especially in remote areas. Nonetheless, fuel-reduction treatments that reduce fire hazard and that persist a few years may be justified by the significant threat of fire in the absence of treatments and the associated values at risk (especially populations of people or rare species). Ultimately, mitigating for the hot and dry climatic conditions that predispose these forests to periodic high-intensity crown fires is a battle not likely to be won by manipulating fuels: Weather drives the stand-replacing fires that periodically occur in these forests. As with spruce-fir forests, fire spread in mixed-conifer forests is constrained primarily by fuel moisture (Rollins and others 2002; Schoennagel and others 2004), which indicates that attempts to manipulate fuels may not reduce the incidence of stand-replacing fires. Such fires are an important part of the mixed-severity fire regime that characterized these systems before Anglo settlement. This regime is at least partially responsible for the high biological diversity of these forests. Attempting to eliminate surface fires or stand-replacing fires likely would cause several species to decline or disappear from these biologically rich forests.

Treating fuels with livestock

Herbaceous plants contribute little to the fuel load. Rather, nearly all the fuel is downed woody material, low-hanging dead branches, and conifer leaves. Thus, removal of fuel via livestock grazing has little potential for reducing fire hazard in mixed-conifer forests.

Recommended treatments

Wildland fire use and reintroduction of periodic surface fires is recommended to manage fuels in this type. High-intensity, stand-replacing fires that would enhance biological diversity are too dangerous to implement as prescribed fires, given contemporary technology. Livestock are ineffective managers of fuel, and mechanical treatments represent a transient, expensive, and likely ineffective subsidy on economic activity in an inappropriate location. In rare cases, fuel-reduction treatments that reduce fire hazard and that persist a few years may be justified by the significant threat of fire in the absence of treatments and the associated values at risk (especially populations of people or rare species).

Ponderosa pine forest

Ponderosa pine forests are found at sites drier than mixed-conifer forests and moister than pinyon-juniper woodlands, though the ecotone between these types may be large in some mountain ranges. Ponderosa pine forests correspond to Kuchler's (1964) Arizona pine forest, and are found on about 8.5 million acres in Arizona and New Mexico (Brown 1982, pp. 43-44). They typically are found at elevations between 6,000 feet and 7,000 feet, and average annual precipitation varies between 15 inches and 28 inches (Brown 1982, pp. 42-48; Wright and Bailey 1982).

The relatively continuous ponderosa pine forest stretching along the Mogollon Rim between northern Arizona and central New Mexico is reportedly the largest ponderosa pine forest in the world (Fulé et al. 2001a). It is the only major commercial forest in the American Southwest. These factors, combined with the proximity of the internationally renowned University of Arizona Laboratory of Tree-Ring Research and the large forestry program at Northern Arizona University, help explain the considerable attention paid to ponderosa pine forests in the region. Southwestern ponderosa pine forests are among the most-studied ecosystems in the world.

The dominant tree species is ponderosa pine, and it varies in density from very open, park-like stands to dense thickets. This variation in stem density often is attributed to human-induced changes in fire regimes, with increased density attributed to the virtual absence of fires since the late 1880s (e.g., Brown 1982; Wright and Bailey 1982; Pollet and Omi 2002). Gambel oak (*Quercus gambelii* Nutt.) and New Mexico locust are common subordinates that may be locally dominant on relatively xeric or rocky sites. On relatively mesic sites, subordinate species include Douglas-fir, southwestern white pine, and quaking aspen.

Fire effects on common trees

Ponderosa pine has many attributes that enhance fire tolerance (Weaver 1967; Boldt and Van Deusen 1974; Schubert 1974; Dieterich 1979; Wright and Bailey 1982; Sackett 1984; Oliver and Ryker 1990). Adult trees have thick, platy bark; the spaces between plates dissipate heat, and the plates exuviate, thus protecting cambium. The relatively open canopy also allows heat to dissipate, thereby minimizing crown scorch. Also, buds are encased in a fascicle of needles that absorbs much heat before the well-insulated buds can be damaged. Ponderosa pine is deep-rooted, even at an early age. In addition to its considerable resistance to fire, ponderosa pine generally recruits readily into mixed-conifer forests after fire. Although ponderosa pine seeds are large and therefore not particularly well dispersed by wind, the relatively patchy character of fires in these forests ensures an adequate supply of seeds from adult trees near recently burned areas. Ponderosa pine seedlings establish readily in mineral soil in these relatively mesic forests. Height growth and diameter growth are rapid, such that trees become large and therefore fire-resistant relatively quickly. Seedlings are moderately susceptible to fire, with mortality strongly dependent on fire intensity. For example, most seedlings less than 4.5 feet tall survived "light" (and presumably low-intensity) surface fires on the Fort Apache Indian Reservation, about half survived "hot" (and presumably high-intensity) surface fires, and about one-fourth survived crown fires (Lindenmuth 1960). This combination of high recruitment, rapid growth, and high resistance makes ponderosa pine particularly well suited to surviving surface fires and establishing in the wake of crown fires.

Ponderosa pine is less likely to establish quickly in these forests than in the more mesic mixed-conifer forests, apparently because of limited soil moisture. Therefore, it has sometimes recruited in region-wide pulses during the last century, and these episodic recruitment events have been linked to periods of above-average precipitation (Cooper 1960). Although ponderosa pine seeds are large and therefore not particularly well-dispersed by wind, seed availability does not seem to constrain recruitment, perhaps because some seed-bearing trees survive even high-intensity crown fires. Thus, the irregular character of high-intensity fires in these forests ensures an adequate supply of seeds from mature trees within or adjacent to burned areas. After seedlings establish, they typically experience rapid height growth and diameter growth. Seedlings are moderately susceptible to fire, with mortality strongly dependent on fire intensity. Unlike conifer species such as white fir and Douglas-fir, ponderosa pine seems largely immune to successive stand-replacing fires. Episodic recruitment, rapid growth, and high resistance to fire makes ponderosa pine particularly well-suited to surviving surface fires and establishing in the wake of crown fires.

Gambel oak (*Quercus gambelii* Nutt.) is a dense, clonal shrub throughout much of the American Southwest, but widely spaced trees of the species are found beneath stands of ponderosa pine. Trees are relatively resistant to low-intensity surface fires; trees top-killed by fire usually resprout, often prolifically (Brown 1958; Tiedemann and others 1987; Fulé, and others 1997). Plants resprout within a few days after they are top-killed by fire and they grow relatively quickly, such that Gambel oak may dominate stands within the first several years after fire.

New Mexico locust responds to fire in a manner similar to its response in higher-elevation forests. Stems are easily killed by fires, but rapid resprouting and recruitment from scarified seeds allows the species to become locally dominant. These patches of New Mexico locust usually give way to conifers within 15-20 years.

Douglas-fir, southwestern white pine, and quaking aspen usually are absent from moderate to xeric ponderosa pine forests, although they are found in relatively mesic ponderosa pine forests. On these relatively mesic sites, response of these species is similar to response in mixed-conifer forests.

Fire regimes and their consequences

Probably no ecosystem is better studied, especially with respect to fire history and ecology, than these forests in this region. Despite this long history of study (or perhaps because of it), there is considerable debate about historical fire regimes. For many years, scientific consensus held that ponderosa pine forests were characterized by high-frequency, low-severity surface fires. Within the last decade, considerable evidence points to the existence of infrequent, high-intensity crown fires (e.g., Shinneman and Baker 1997; Brown and others 1999; Johnson and others 2001, Ehle and Baker 2003). It now seems that most stands were characterized by a mixed regime, with a regime of frequent surface fires overlain by infrequent, stand-replacing fires. A holistic perspective that incorporates weather, climate, and site-specific attributes of stands can explain much of the variability masked by contentious debate.

Although fire occurrence and behavior are strongly dependent on weather conditions (Bessie and Johnson 1995; Shinneman and Baker 1997; Pierce and others 2004), density and cover of ponderosa pine may affect fire frequency, seasonality, and intensity. Historically, a positive feedback probably was maintained on many sites, with low stem density contributing to high-frequency, low-intensity fires during the early summer (coincident with the southwestern "monsoon"). This fire regime probably maintained relatively open, park-like stands of widely scattered pine trees throughout the American Southwest for many years, and perhaps for centuries at a time on some sites. Closed-canopy stands doubtless established on relatively mesic sites and during periods of above-average precipitation (e.g., Humphries and Bourgeron 2003). The "ladder fuels" in these stands could carry fire into tree canopies, where fire behavior is largely controlled by foliar moisture (Agee and others 2002; Bilgili 2003). Thus, mixed fire regimes could have occurred on most sites currently dominated by ponderosa pine, and the transition from a regime dominated by high-frequency, low-intensity surface fires to one dominated by low-frequency, high-intensity crown fires could result from modest variations in precipitation over relatively short periods of time. For example, a decade-long period of above-average precipitation could engender rapid recruitment and growth of pine trees while also acting to reduce fire occurrence and spread. A subsequent decade-long period of below-average precipitation could lower fuel moisture, thereby triggering widespread stand-replacing fires. Individual fires tend to be highly variable in behavior, and heterogeneity in fuels, microclimate, and myriad other factors explain why even fires in very dry, closed-canopy forests rarely behave only as crown fires (Kafka and others 2001). A large-scale, long-term perspective that incorporates spatial and climatic variability thus helps clarify a debate arising from apparently conflicting data.

Stand-replacing fires probably have become more common since livestock grazing and fire suppression largely eliminated low-intensity surface fires within the last century or so. However, stand-replacing fires likely were common during the "Medieval Climatic Anomaly" (MCA) of ca. 1,050 – 650 years ago (Pierce and others 2004); contemporary and likely near-future climatic patters are similar to the MCA (Crowley 2000; Bradley and others 2003).

Gambel oak, New Mexico locust, or both species typically dominate the post-fire overstory, often for several decades. When heavy rains follow stand-replacing fires, significant soil loss may occur, especially on sites with relatively steep slopes. On these sites, shrub thickets may persist for several centuries, even in the face of Herculean attempts to reintroduce ponderosa pine forests. The inherent productivity of these sites is compromised by the loss of soil, to which cultural inputs such as herbicide application and reforestation are no match.

Mechanical treatment of fuels

Harold Weaver's (1951, 1967) pioneering research convincingly demonstrated that high-intensity crown fires in dense thickets can be transformed to surface fires as they pass into open, park-like stands of ponderosa pine. The low-density stands studied by Weaver were maintained by a combination of intensive timber removal and frequent fires. Reintroduction of cool-season (i.e., spring, autumn) prescribed fire alone usually is not sufficient to create stands that are resistant to crown fire (Sackett and others 1996; Fernandes and Botelho 2003).

Model simulations of fire behavior in ponderosa pine of northwestern Arizona indicate that mechanical treatments reduced intensity, flame length, and crown scorch relative to untreated stands (Fulé and others 2001b). Mechanical treatments decreased the likelihood of crown fire in this study, a finding that corresponds to model results for ponderosa pine forests in Arizona (Covington and others 1997) and Montana (Scott 1998), as well as lodgepole pine forests (Kalabokidis and Omi 1998) and mixed-conifer forests (Graham and others 1999; Fiedler and others 2003). However, mechanical removal of trees may not substantively reduce the risk of crown fires unless these mechanical treatments are coupled with periodic prescribed fires; thus, “stand-alone” mechanical treatment parallels “stand-alone” treatment with cool-season fire. The combination of mechanical treatment and fire was the technique advocated by Weaver. Subsequent modeling efforts indicate that slash produced by mechanical treatments (conducted by themselves) increases the probability of fire, largely because fire behavior typically is influenced by the fine fuels that are left on-site in the wake of mechanical operations (Fulé and others 2001a; Graham and others 2004). For the same reason, mechanical treatments that remove relatively large trees do not decrease the potential for crown fire (e.g., Kalabokidis and Omi 1998; Graham and others 1999). Empirical evidence indicates that fine fuels left on-site after mechanical treatments may increase the intensity of subsequent fires for at least three years (Graham and others 2004).

Treating fuels with livestock

The high cover and density of ponderosa pine trees in most of the region’s ponderosa pine forests preclude development of a significant herbaceous understory. To a great extent, pine needles and stems fuel contemporary fires, either as surface fires or crown fires. Livestock have minimal impact on these fuels. Livestock grazing will reduce the probability of fire spread through ponderosa pine forests with a well-developed grass layer and pine savannas. However, reducing fire occurrence and spread in these systems is counter to the goal of maintaining high levels of biological diversity of native species, and it likely will not reduce the probability of high-intensity crown fires.

Recommended treatments

Wildland fire use and also mechanical treatment of fuels coupled with reintroduction of summer prescribed fires are recommended for many sites in this type. Mechanical fires in the absence of subsequent fires or prescribed fires in the absence of preceding mechanical treatments may be useful in isolated areas, depending on prior land use and contemporary structure. Although high-intensity, stand-replacing fires likely should be mixed with surface fires of lower intensity, the former fires are too dangerous to implement as prescribed fires in most areas. Prescribed fires during fall, spring, or winter are not recommended because of declines in biological diversity that likely would result from application of fires during seasons that are "novel" to species in this region (see section titled "Fire management in altered contemporary ecosystems"). Livestock may reduce fuels necessary for fire spread (and therefore reduce options for reintroduction of seasonally appropriate fires).

Evergreen oak woodland

Evergreen oak woodlands (the "encinal" of Shreve 1915) cover about 1.5 million acres in Arizona and New Mexico, primarily along the Mogollon Rim and in the mountains of southeastern Arizona and extreme southwestern New Mexico. These woodlands correspond roughly to the oak-juniper type of Kuchler's (1964) classification system, although Kuchler does not acknowledge the extensive stands of oaks in which juniper trees are absent. These woodlands typically are found at elevations of 5,000 feet to 6,500 feet, above semi-desert grassland and below ponderosa pine forest (with which evergreen oak woodlands mix and co-mingle). Annual precipitation averages 15 inches to 25 inches (Brown 1982, pp. 59-65).

Dominant trees in Arizona woodlands include Emory oak (*Quercus emoryi* Torr.), Arizona white oak (*Q. arizonica* Sarg.), and south of the Gila River, Mexican blue oak (*Q. oblongifolia* Torr.). Emory oak and gray oak (*Q. grisea* Liebm.) are dominant in southeastern Arizona and southern New Mexico, and silverleaf oak (*Q. hypoleucoides* Camus) and netleaf oak (*Q. rugosa* Nee) dominate at relatively mesic high-elevation sites. Warm-season (C_4) grasses dominate the understory throughout the region, particularly at the lower and drier end of this type as it grades into oak savanna and then semi-desert grassland.

Fire effects on common trees

Although fire effects have been little studied in this system, the response to fire appears to be broadly similar for all species of evergreen oak. All species experience high fire-induced mortality as seedlings. Saplings have adequate root systems to resprout when burned, and the proportion of trees that resprouts after a fire varies between about 20% and 80%; survival of saplings apparently increases with increasing soil moisture availability and decreases with increasing fire intensity, tree size (once trees progress beyond the seedling stage, post-burn survival declines with increased tree size), and soil depth (which facilitates rapid growth of trees) (Johnson and others 1962; Cable 1975; Niering and Lowe 1984; Davis 1987; Babb 1991; Caprio and Zwolinksi 1992, 1995; Ffolliott and Bennett 1996; Barton 1999). Other factors, such as fire frequency and season, almost certainly affect post-fire survival of these species, but they have not been investigated.

Fire regimes and their consequences

Fire regimes in oak woodlands have been the source of little speculation and even less data. Ecology of oak woodlands has been understudied relative to other systems in the region, and fires leave little historical evidence. Fires rarely leave fire scars on the dominant tree species. In addition, stem age of these species is neither easily nor reliably determined via dendrochronological techniques. Finally, stem age does not indicate age of establishment because many individuals of all southwestern species resprout when top-killed. Thus, fire history can not be inferred from stand structure (McPherson 1997, McPherson and Weltzin 2000).

Savannas at the lower and drier margin of oak woodlands probably burned every 10-20 years on average, similarly to adjacent grasslands. Variability doubtless was quite high, with fires occurring as frequently as every two years (Kaib and others 1996). Interruption of frequent fires since Anglo settlement likely contributed to the development of stands with increased stem densities within the last century (e.g., McPherson and others 1993), a trend that has been documented for other savannas throughout the world (Archer 1994; McPherson 1997). These low-intensity surface fires likely swept into the adjacent closed-canopy woodlands during some years.

It seems unlikely that closed-canopy woodlands, with their lack of herbaceous understory, would have supported surface fires at the high frequency found in adjacent savannas. However, these trees shed all their leaves each year a few months before the southwestern "monsoon" as new leaves expand to replace the leaves from the previous year. These leaves decompose slowly and they are well-dried between leaf senescence (ca. April) and the first thunderstorm (typically, early July). Thus, the abundant supply of leaves may have served as volatile fuel, especially in concert with fine woody fuels that accumulated over time. The few months of consistently hot, dry weather between leaf senescence and ignition sources may have created conditions amenable to high-intensity crown fires. Under this scenario, trees that survived the fire would resprout and seedlings would recruit from acorns into the bare spaces. Thus, these woodlands may have been characterized by periodic low-intensity surface fires and less frequent high-intensity crown fires. In the absence of fire-history data in these systems, this scenario remains speculative; other reasonable scenarios are equally suspect.

Mechanical treatment of fuels

Forecasting the outcome of fuels treatments is perilous considering the considerable uncertainty that surrounds the determination of historical fire regimes. Nonetheless, separate scenarios are considered for savannas and more mesic closed-canopy woodlands.

In oak savannas, the herbaceous layer controls fire occurrence and spread, and mechanical removal of woody fuels has negligible impacts on fire regime. In former grasslands and savannas, reducing cover of woody plants likely will allow herbaceous plants to increase in abundance and therefore facilitate fire spread. Closed-canopy stands can be transformed back into grasslands or savannas, but only with enormous cultural inputs (e.g., bulldozers and herbicide). These treatments will not persist indefinitely: Oak trees resprout and new trees establish from acorns, especially on deep-soiled sites.

Removal of oak trees from relatively mesic closed-canopy woodlands may reduce the likelihood of high-intensity crown fires. The persistence of such fuel-reduction treatments has not been evaluated, and mechanical removal of woody fuels is economically expensive, especially in remote areas. Such expensive and possibly transient fuel-reduction treatments may be difficult to justify, given that the threat of high-intensity fire is unknown.

Treating fuels with livestock

Livestock grazing reduces the probability of fire occurrence and spread in southwestern savannas, as described by Aldo Leopold (1924) more than 80 years ago. The idea that "grazing prevents blazing" is a simplistic overstatement—many wildfires start on or spread to areas grazed by livestock—but the fire regime in these systems is strongly influenced by the amount and distribution of herbaceous plants, which may be significantly altered by livestock. Thus, using livestock in oak savannas can reduce the probability of fire. However, reducing fire occurrence and spread in these savannas is counter to the goal of maintaining high levels of biological diversity of native species.

In more mesic oak woodland, herbaceous plants contribute little to the fuel load and therefore to fire spread. Thus, removal of fuel via livestock grazing has little potential for altering fire hazard; even long-term increases in woody plants that might result from removal of herbs have unknown, and likely minimal, impacts on flammability of these systems. An exception is use of goats to reduce cover of woody plants that comprise the dominant fuels in these systems. However, the resulting reduction in woody plants may stimulate an increase in herbaceous plants, which could enhance the probability of surface fires.

Recommended treatments

Given the nearly complete lack of knowledge about fire regimes, especially in mesic woodlands, recommendations are made tenuously and should be interpreted with caution. Treatment of fuels with machines or livestock appears unwarranted and detrimental to biological diversity. Wildland fire use and prescribed fires during summer, particularly in savannas, likely would not reduce, and may enhance, biological diversity. The persistent impact of these (and other fires) on fuels is unknown. Prescribed fires during other seasons, except in small areas or to reduce fuels in preparation for subsequent fires, are not recommended because of the likely consequent decline in biological diversity.

Pinyon-juniper woodland

Pinyon-juniper woodlands cover nearly 40 million acres in Arizona and New Mexico, primarily north of the Mogollon Rim. Pinyon and juniper trees dominate many relatively xeric sites north of the Mogollon Rim, where they correspond to the juniper-pinyon type of Kuchler's (1964) classification system. South of the Rim, this woodland is found in a band of vegetation between about 4,000 feet and 6,000 feet in the sky islands; here they comprise part of the oak-juniper type of Kuchler's (1964) classification system.

South of the Rim, they are the lowest ecosystems in the southwestern United States dominated by conifers, and they typically occur just above semi-desert grasslands. These woodlands average between 10 inches and 20 inches of annual precipitation, with most of the precipitation occurring as snow in the northern portion of the region and nearly all occurring as rain in the sky islands of southern Arizona and southern New Mexico (Brown 1982, pp. 52–65).

Common tree species include Rocky Mountain (synonymous with doubleleaf) pinyon (*Pinus edulis* Engelm.), singleleaf pinyon (*P. monophylla* Torr. & Frem.), Rocky Mountain juniper (*J. scopulorum* Sarg.), Utah juniper [*J. osteosperma* (Torr.) Little], and one-seed juniper [*Juniperus monosperma* (Engel.) Sarg.] north of the Mogollon Rim and Mexican pinyon (*Pinus cembroides* Zucc.), one-seed juniper, and alligator juniper (*Juniperus deppeana* Steud.) south of the Rim. Pinyon trees typically are restricted to the most mesic sites in these woodlands, and are therefore absent from low-elevation sites where pinyon-juniper woodland contacts grassland. Rocky Mountain juniper is found on more mesic sites than the remaining juniper species. A well-developed shrub understory dominated by big sagebrush (*Artemisia tridentata* Nutt.) often is present beneath pinyons and junipers on the Colorado Plateau. Cool-season (C_3) grasses dominate the understory on the Colorado Plateau, giving way to warm-season (C_4) grasses south of the Mogollon Rim and east of the continental divide in New Mexico.

Fire effects on common trees

Individual pinyon trees typically possess few adaptations to fire, which helps explain why they often are restricted to the most mesic sites in pinyon-juniper woodlands. Before Anglo settlement, individual pinyon trees in juniper-dominated stands likely escaped fires because they occupied rocky areas with little herbaceous biomass. The same principle apparently applied to entire stands of pinyon-juniper woodland, especially along the rocky escarpment of the Mogollon Rim and at scattered locations on the Colorado Plateau: These stands rarely burned, which allowed fire-intolerant pinyon trees to live for several centuries (e.g., Tress and Klopatek 1987; Kruse and Perry 1995; Floyd and others 2000; Romme and others 2003).

As seedlings, all juniper species except alligator juniper are very susceptible to fire, even low-intensity fires in grasslands and savannas. However, junipers become very resistant to surface fires when they reach about 4.5 feet in height. Alligator juniper, which resprouts prolifically from the base when top-killed, is particularly well-adapted to fire (Wright and Bailey 1982). Juniper species may colonize rapidly into recently burned sites, areas dominated by warm-season grasses, or sites dominated by sagebrush adjacent to pinyon-juniper woodlands. Colonization is facilitated by above-average precipitation and removal of herbs by livestock. Both factors alleviate constraints on water uptake by juniper seedlings, the former directly and the latter by removing plants that otherwise interfere with juniper establishment (Wright and Bailey 1982, McPherson 1997).

Fire regimes and their consequences

Before Anglo settlement, pinyon-juniper woodlands were characterized by a mixed fire regime. Low-intensity surface fires were carried by herbaceous fuels, primarily grasses, in relatively xeric juniper savannas. Stand-replacing fires of moderate intensities characterized more mesic sites, especially on the Colorado Plateau where shrubs were common in the understory. These fires typically were ignited by lightning during late June and early July thunderstorms at the leading edge of the southwestern "monsoon." The most mesic pinyon-juniper woodlands rarely burned. Thus, the pre-Columbian fire regime of pinyon-juniper woodlands can be classified into three distinct types (Romme and others 2003).

The most xeric pinyon-juniper stands burned every 5 to 20 years, with considerable variability around mean frequency, before Anglo settlement. Time between fires depended strongly on post-fire recovery of grasses. Fires in these savannas were constrained by accumulation of herbs, such that they occurred most frequently on relatively productive sites (e.g., those with high precipitation and deep, well-drained soils) and less frequently on less-productive sites or during periods of below-average precipitation. Fires spread quickly, and burned until they were doused by "monsoonal" storms or encountered fuel breaks such as rivers and escarpments. Various juniper species invaded these grassland and savanna sites during the last century, presumably as a result of livestock grazing and the attendant reduction in fire frequency (Wright and Bailey 1982, McPherson 1997). At the relatively low densities of woody plants that characterize juniper savannas, low-intensity surface fires still may kill most juniper plants and thus reduce their density, as occurred before Anglo settlement. However, juniper trees grow quickly enough, especially in the presence of livestock and the absence of fire, that fires will no longer kill juniper plants a few years post-establishment. In many cases, fires have been absent long enough from former grassland and savannas to allow the formation of closed-canopy stands with little herbaceous understory. The closed-canopy stands of juniper that now dominate many sites rarely burn because fine fuel is inadequate to support fire spread even during hot, dry, windy conditions. The sparse herbaceous layer is inadequate to support fire spread in the understory, and trees are too far apart to support crown fires. Thus, livestock grazing and fire suppression apparently have transformed millions of hectares of grassland and savannas into pinyon-juniper woodlands with attendant alterations in ecosystem function. Considerable effort and much progress have been made in the application of prescribed fires to these systems.

Pinyon-juniper woodlands on more mesic sites, especially those with a conspicuous understory of big sagebrush on the Colorado Plateau, were characterized by a fire regime that was intermediate in frequency and intensity. Pinyon trees often were present and trees were sufficiently dense to support the spread of high-intensity crown fires during particularly hot, dry, windy conditions. In these stands, fires were constrained by a combination of fuel accumulation and fuel moisture, but they occurred every few decades. They were fueled primarily by woody plants, so that the probability of fire was low until woody plants established and grew to "fill in" after the last fire. The probability of fire increased as fuels accumulated in the post-fire period. A few decades after a fire, the availability of fuel corresponded with an ignition source (usually lightning) and low fuel moisture associated with the hot, dry weather that characterizes the region in May and June. Under these conditions, fire was (and is) inevitable. The pre-settlement fire regime has been maintained on many of these sites. On others, interruption of the fire regime is a temporary phenomenon that is maintained only by the considerable expenditure of fire-suppression funds. Ultimately, these stands will burn. Fire intensity undoubtedly increases with increased time since fire because the woody fuels that support fire spread in these systems continue to accumulate. Thus, the artificial and temporary removal from these systems spawns less frequent but more intense fires in these systems. Such fires may change soil chemistry and soil structure, thereby causing accelerated erosion (e.g., DeBano 2000; Robichaud and others 2000; Sampson and others 2000; Ritsema and Dekker 2003).

Relatively tall trees and very infrequent fires characterize the most mesic pinyon-juniper woodlands. When these woodlands burn, fire intensity is adequately high to consume much of the aboveground biomass. Given the centuries-long fire-return intervals that typified these forests before Anglo settlement, it seems reasonable to assume that a century of fire suppression has had minimal impact on disturbance dynamics or stand structure. Thus, the contemporary fire regime probably is very similar to the pre-settlement regime. Considering the susceptibility of dominant tree species to fires (especially pinyon) and the fuel and weather conditions associated with historical and contemporary fires, introducing prescribed fires into mesic pinyon-juniper woodlands seems very difficult to accomplish.

Although the fire regime apparently has been largely unaffected by Anglo settlement in the most mesic pinyon-juniper woodlands, the impacts of contemporary fires probably will differ from those of pre-settlement fires. Regional warming and drying as a result of global warming probably affect tree recruitment patterns. Specifically, establishment of pinyon likely will decline or will be restricted to years with above-average precipitation. Stand-replacing fires in these systems, particularly in sequence, might engender a region-wide reduction in pinyon and a concomitant increase in juniper, sagebrush, and other species typically found on more xeric sites. Rapid warming and drying likely will elevate the likelihood of fire occurrence and intensity in these woodlands, similar to patterns observed (Wotton and others 2003) and expected (Baker 2003; Flannigan and others 2003; Miller 2003) in coniferous forests of North America.

Mechanical treatment of fuels

In juniper savannas, the herbaceous layer controls fire occurrence and spread. Mechanical removal of woody fuels has negligible impacts on fire regime. In former grasslands and savannas that have been transformed into closed-canopy woodlands within the last century, reducing cover of woody plants likely will allow herbaceous plants to increase in abundance and therefore facilitate fire spread. With sufficient expenditure of capital, closed-canopy stands can be transformed back into grasslands or savannas. Many examples of these “type conversions” have occurred in Arizona and New Mexico within the last half-century. These treatments usually are not persistent, especially in the presence of livestock: junipers begin invading most sites almost immediately. Fiscally, the most reasonable approach to maintaining these savannas is to reintroduce fire at a frequency and season appropriate to the site (i.e., early summer every 5-20 years). This will nearly always require removal of livestock for one to several years to facilitate recovery of herbaceous biomass and therefore allow fire spread.

Removal of woody plants from relatively mesic pinyon-juniper woodlands, such as those with a conspicuous understory of big sagebrush, may reduce the likelihood of fire spread. The persistence of such fuel-reduction treatments has not been evaluated, but it probably does not exceed five years (e.g., Fernandes and Botelho 2003). Furthermore, mechanical removal of woody fuels is economically expensive, especially in remote areas. Nonetheless, expensive and transient fuel-reduction treatments may be justified by the significant threat of fire in the absence of treatments and the associated values at risk (e.g., housing developments, populations of rare species). Ultimately, however, mitigating for the hot and dry climatic conditions that predispose these woodlands to high-intensity, stand-replacing fire involves accepting one of two probable scenarios: (1) short-term reduction in the probability of fire, or (2) a commitment to repetitive, expensive fuel-reduction treatments.

In summary, removal of woody plants from the most mesic pinyon-juniper woodlands probably will not alter fire occurrence or spread significantly. The fire regime in these woodlands is determined primarily by weather: Fire spread is constrained largely by fuel moisture, not by fuel continuity. Climatic conditions increasingly favor fires, which indicates that attempts to manipulate fuels likely will not reduce the incidence or spread of fires.

Treating fuels with livestock

Livestock grazing reduces the probability of fire occurrence and spread in southwestern savannas. Thus, using livestock in juniper savannas can reduce probability of fire. However, reducing fire occurrence and spread in these savannas is counter to the goal of maintaining high levels of biological diversity of native species.

In more mesic pinyon-juniper woodlands, herbaceous plants contribute little to the fuel load and therefore to fire spread. Thus, removal of fuel via livestock grazing has little potential to reduce fire hazard. An exception is use of goats to reduce cover of woody plants that comprise the dominant fuels in these systems. Goats have been used to reduce fire hazard in California shrublands, but this approach seldom has been attempted in the American Southwest (but see research on goats in chaparral communities by Knipe 1983 and Severson and DeBano 1991).

Recommended treatments

Treatment of fuels with machines or livestock appears unwarranted, expensive, ineffective, and likely detrimental to biological diversity. In rare cases, fuel-reduction treatments that reduce fire hazard and that persist a few years may be justified by the significant threat of fire in the absence of treatments and the associated values at risk (especially populations of people or rare species): Fuelwood projects in small but economically important areas might be useful in this regard. Prescribed fires during summer, particularly in savannas, likely would not reduce, and may enhance, biological diversity. Prescribed fires during other seasons are not recommended because of the likely consequent decline in biological diversity.

Southwestern grassland

Grasslands cover about five million acres in Arizona and New Mexico. These grasslands are scattered throughout the two-state region and are dominated by a diverse suite of species. As a result, they typically are classified into several different types. For example, Kuchler (1964) recognizes grama-buffalograss grasslands in eastern New Mexico, wheatgrass-needlegrass grasslands on the Colorado Plateau, and grama-tobosa prairie in southeastern Arizona and southern New Mexico. Various authors split the grasslands of southeastern Arizona and southern New Mexico into two types: semi-desert grassland and plains grassland (e.g., Brown 1982). The semi-desert grassland can be further divided into grasslands dominated by black grama (*Bouteloua eriopoda* Torr.) and those in which black grama is uncommon or absent. Despite the variety of physiographic settings and consequent diversity of dominant species, issues surrounding fuels management are quite similar in these grass-dominated systems. Therefore, they are combined into a single section for this discussion.

Grasslands of the Colorado Plateau typically receive between 7 inches and 12 inches of annual precipitation, a significant majority of which falls as snow. Annual precipitation in the plains grasslands and semi-desert grasslands of southeastern Arizona and southern New Mexico averages between 10 inches and 18 inches; these grasslands receive about half their precipitation in the form of summer thunderstorms. Shortgrass prairie in the eastern third of New Mexico receives a similar amount of precipitation, but less of it falls during summer. South and east of the Mogollon Rim, grasslands typically are found between 3,500 feet and 5,500 feet and are bordered by oak- or juniper-dominated woodlands at higher elevations and by xeric communities dominated by woody plants at lower elevations (e.g., desert scrub or semi-desert shrub/grassland) (Brown 1982, pp. 115-131).

Cool-season (C_3) grasses mix with warm-season (C_4) grasses on the Colorado Plateau west of the Continental Divide. This grassland represents the southern extension of the sagebrush-grass system that dominates the Great Basin, and the rich array of grasses reflects species found in adjacent sagebrush-dominated sites: Galleta [*Hilaria jamesii* (Torr.) Benth.], several species of grama (*Bouteloua*), dropseed (*Sporobolus*), bluegrass (*Poa*), and needle-and-thread (*Stipa*) are a few of the common taxa. South and east, cool-season grasses may be present but warm-season grasses assume dominance. Grasses of the following genera are particularly prevalent: grama, three-awns (*Aristida*), muhly (*Muhlenbergia*), and dropseed. Floodplains usually are dominated by tobosa [*Hilaria mutica* (Buckl.) Benth.,] and sacaton (*Sporobolus wrightii* Munro).

Fire effects on woody plants and grasses

The great variety of species and mixtures of species precludes consideration of a species-by-species discussion. Rather, this section focuses on general relationships between woody plants and grasses, and the impact of fire on these relationships. Shifts in vegetation physiognomy have important implications for management and land use: They affect soil, water, wildlife, livestock, fuelwood, recreation, carbon sequestration, non-methane hydrocarbon emissions, albedo, evapotranspiration, surface roughness, and boundary-layer dynamics, among other factors (McPherson and Weltzin 2000).

Biomass and cover of woody plants and grasses are inversely related, and grass biomass generally declines rapidly with increasing cover of woody plants. Historically, recurrent fires reduced woody plant cover, thereby favoring grasses. Grasses provided fuel for recurrent fires, such that a positive-feedback cycle was sustained until the cycle was interrupted: Livestock grazing and fire suppression favored woody plants in myriad ways. The virtual absence of fires from these systems since the late 1800s has, in many cases, facilitated a shift from grassland to shrubland. These shrublands typically have a low probability of fire spread because fine fuels are sparse and discontinuous.

Summer fires were (and probably are) particularly important for sustaining grasses at the expense of woody plants. Although perennial plants, including grasses, are susceptible to mortality from early-summer fires, the effects of fires during this season are more obvious and more persistent than fires during cooler times of the year (Glendening and Paulsen 1955; Tschirley and Martin 1961; Cable 1965, 1967, 1973; Pase 1971; Martin 1983). Many woody plants may require a decade or more to recover from fires, but grasses generally recover within three years (Wright and Bailey 1982; McPherson 1995).

Fire regimes and their consequences

Historical accounts suggest that extensive fires occurred frequently in most of the region's grasslands. Mean time between fires probably was 5-15 years; there was considerable variability around the mean on a site, such that fire-free episodes likely ranged from 2 to 30 years (Humphrey 1958; Humphrey and Mehrhoff 1958; Wright and Bailey 1982; McPherson 1995). These fires typically occurred in late June or early July when the first summer thunderstorms moved into the region after the extended hot, dry period in May and June.

Many woody plants found in these grasslands are susceptible to fire, especially as seedlings (Bock and Bock 1992), and few of the woody plants resprout if they are burned as seedlings (Glendening and Paulsen 1955; Wright and Bailey 1982). Fires also kill seeds of woody plants found on the soil surface (Cox and others 1993). Many of the region's woody plants produce seeds only after they are at least 10 years old (Humphrey 1958; Burns and Honkala 1990a, 1990b), which suggests that relatively frequent fires helped prevent the establishment of woody plants. In contrast, most grasses experience a period of reduced productivity shorter than three years (Wright and Bailey 1982). These differences in post-fire response likely gave grasses a competitive advantage and helped prevent shrub encroachment before Anglo settlement.

When livestock were introduced throughout the region in the late 1800s, the concomitant reduction in fine fuels reduced the probability of fire spread, thus contributing to a rapid decline in fire occurrence and widespread dominance by woody plants (McPherson 1995). On many sites, woody plants now interfere with grass production to the extent that fires rarely spread, even under climatic conditions that are extremely hot, dry, and windy.

Fire is not crucial for preventing shrub establishment in some plains grassland sites, especially relative to nearby semi-desert grassland. Woody plants associated with plains grassland tend to be smaller than those associated with semi-desert grassland, perhaps because of colder temperatures or edaphic constraints in plains grasslands. As a result, the positive relationship between fire occurrence and grasses is less obvious on plains grassland than in most other southwestern grasslands (Dick-Peddie 1993; Paysen and others 2000).

In contrast to most other grasslands of the region, historical fires apparently were uncommon in grasslands dominated by black grama. There is a paucity of historical evidence for widespread fires in southern New Mexico where black grama dominated semi-desert grasslands; in addition, the slow post-fire recovery of this species, particularly when fires are followed by below-average precipitation, suggests that it could not dominate sites subject to frequent fires (Buffington and Herbel 1965).

Nonnative perennial grasses, especially Lehmann lovegrass (*Eragrostis lehmanniana* Nees), have invaded large portions of semi-desert grassland in southern Arizona during the last 60 years (Ruyle and others 1988; Cox and others 1990; Anable and others 1992). Prescribed fires, which typically are conducted during March and April, favor Lehmann lovegrass, thereby perpetuating the establishment and spread of these grasses. This apparently produces a positive-feedback cycle in which nonnative grasses increase after fire. These grasses typically produce more biomass than native grasses, especially during years of below-average precipitation, and they have a high C:N ratio that makes them slow to decompose. These systems, in which fire spread is limited by fuel abundance and continuity, are therefore subject to more frequent fires when nonnative grasses become dominant (McPherson and Weltzin 2000). However, this positive-feedback cycle has largely failed to overcome the combination of livestock grazing, woody plant dominance, effective fire suppression, and fragmentation from roads and other developments that have contributed to regional reductions in fuel abundance and continuity. As such, fires now occur very rarely on most grassland sites.

The infrequent fires that occur in these grasslands usually burn during two distinct seasons. Prescribed fires usually are conducted during March and April, when relatively cool temperatures facilitate control. Wildfires—and rarely, prescribed fires—may occur in late June, which corresponds to the historical fire season. Effects of spring fires are largely unknown, especially for the many ecosystem functions and unnoticed species that are rarely monitored. The primary exception concerns a dominant nonnative grass in these systems: Prescribed fires conducted during March and April favor Lehmann lovegrass, thereby perpetuating the establishment and spread of this species. Impacts of summer fires, including prescribed fires, demand further study throughout these grasslands. In the absence of further study, prescribed fires should augment wildfires during the critical summer season; these fires undoubtedly will enhance long-term preservation of biological diversity because they correspond to the fire season with which native species evolved.

Mechanical treatment of fuels

The herbaceous layer controls fire occurrence and spread in grasslands, and recurrent fires are at least partially responsible for maintenance of grassland physiognomy in most grasslands. Thus, mechanical removal of fuels is not a rational approach across large areas, although mowing may be useful to reduce the probability of fire in small areas (e.g., near structures) for short periods of time. Even if it were possible from a logistical perspective to mechanically remove herbaceous plants, this action probably would not significantly alter fire occurrence or spread for more than the one or two years that would be required for grasses to regrow.

Treating fuels with livestock

Livestock grazing reduces the probability of fire occurrence and spread in southwestern grasslands. Thus, using livestock in these systems can reduce probability of fire. However, reducing fire occurrence and spread in these savannas is counter to the goal of maintaining high levels of biological diversity of native species.

Recommended treatments

Treatment of fuels with machines or livestock appears unwarranted, expensive, ineffective, and likely detrimental to biological diversity. In rare cases, such treatments that reduce fire hazard and that persist a few years may be justified by the significant threat of fire in the absence of treatments and the associated values at risk (i.e., structures). Prescribed fires during summer almost certainly will enhance biological diversity. Prescribed fires during other seasons are not recommended because of the likely consequent decline in biological diversity.

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